LOW-HEAD DAMS ALTER STREAM PHYSICOCHEMICAL CONDITIONS AND LEAF LITTER DECOMPOSITION

A Thesis
by
Beverly Elyse Russing

Submitted to the Graduate School
Appalachian State University
in partial fulfillment of the requirements for the degree of
MASTER OF SCIENCE

May 2015
Department of Biology

LOW-HEAD DAMS ALTER STREAM PHYSICOCHEMICAL CONDITIONS AND LEAF LITTER DECOMPOSITION

A Thesis
by
BEVERLY ELYSE RUSSING
May 2015

APPROVED BY:

_____
Dr. Michael M. Gangloff
Chairperson, Thesis Committee

_____
Dr. Lynn M. Siefferman
Member, Thesis Committee

_____
Dr. Michael D. Madritch
Member, Thesis Committee

_____
Dr. Susan L. Edwards
Chairperson, Department of Biology

_____
Dr. Max C. Poole
Dean, Cratis D. Williams Graduate School

Abstract

Small Low-Head Dams Alter Physicochemical Conditions and Leaf Litter Decomposition

Beverly Elyse Russing
B.S., Appalachian State University
M.S., Appalachian State University

Chairperson: Dr. Michael M. Gangloff, Ph.D

Changes to land use in headwater catchments may impact stream invertebrate communities and ecosystem functions. Increasingly, dam removals are often part of stream restoration projects that focus on improving water quality and aquatic ecosystem connectivity. Anthropogenic barriers including mill dams alter stream habitat conditions and influence benthic communities and stream ecosystem functions. I conducted two field experiments to better understand the impacts of small impoundments on leaf decomposition, a key biologically-mediated ecosystem process. In the first experimet, I deployed leaf packs up- and down-stream of a relict, breached, and intact dam and removed them after 8 weeks. I calculated decomposition rates and quantified invertebrate community structure at each site. Decomposition rates were highest just downstream from the intact dam. Furthermore, I observed a significant negative correlation between conductivity and the rate of decomposition across sites. Changes in chemical factors such as DO and pH near intact dams may influence macroinvertebrate assemblages and leaf decomposition. Next, I conducted a follow up study to document the effects of altered physicochemical conditions on drivers of decompositon. I tested whether macroinvertebrate shredding or microbial colonization was

driving leaf breakdown rates in the presence of a small dam. Leaf packs were either exposed or enclosed in 16 x 22 cm fine mesh (0.5 mm) bags and placed at 4 sites associated with an intact dam for 12 weeks. I calculated decomposition rates and quantified invertebrate communty structure at each site. Decomposition rates were highest when macroinvertebrate shredders were present and had access to the leaves. Macroinvertebrates are usually uncommon or absent within an impoundment yet, at our impounded sub-reach, leaf packs in fine mesh bags had a significantly higher rate of decomposition than exposed leaves. The highest rate of leaf decomposition for both experiments was in the tailwaters of an intact dam. These data suggest that some intact low-head dams may improve habitat conditions for benthic macroinvertebrates while increased nutrient retention within the impoundment may increase biofilm accumulation and microbial decomposition. Moreover, they suggest that criteria currently used to evaluate the effects of dams on stream ecosystems should include a more wholistic understanding of how dam presence and removal might influence stream communities and ecosystem function.

Acknowledgements

I would like to first and foremost thank the Office of Student Research, the Gangloff lab and the Department of Biology at Appalachian State University for funding this project and for logistical support. I would also like to thank Dr. Gangloff and Dr. Siefferman for giving me such an amazing opportunity and allowing me to be a part of their lab. Without their guidance and support, my research master's education would never have been possible. I would also like to thank Dr. Madritch for his contributions to my education and research over the years and for being such an amazing mentor. A special thanks to Michael Perkins, Jordan Holcomb, Michael Thomas, Worth Pugh, Morgan Harris, Laura Boggess, Caroline Hatch, Marco Enriquez, Gary Pandolfi, and the entire ecology class (Fall '12 & 13) for their amazing field and laboratory assistance.

I could have never made it through these last few years if it weren't for the help of my family. Thank you to my mother and father, Caroline and Michael, for making this all possible and supporting me in all areas of my life. Thank you to my brother, Loyd, for always believing in me and giving me the courage to always seek more. You have been such an amazing role model and best friend. I would like to acknowlege my entire family for all of the sacrifices they've had to make in order for me to continue my education.

Finally, I am grateful to Michael Perkins and Carra Parker for years of support and friendship. I couldn't have made it through this without you and it wouldn't have been nearly

as much fun. You both mean the world to me and it has truly made this whole experience worthwhile. Thanks to everyone in the Gangloff and Siefferman Lab past and present.

Dedication

I dedicate this thesis to my brother Loyd J. Russing

Table of Contents

List of Tables

List of Figures

Foreword

The research outlined in this thesis will be submitted to the peer-reviewed journal *Hydrobiologia*. The body of this thesis has been prepared according to the style and formatting requirements for publication in this journal.

Introduction

Dams are one of the most widespread examples of human impacts on streams and affect >1 x 10^6 km of river in the United States alone (Poff et al., 1997). Dams were first constructed for grain milling, flood control, and hydroelectric power generation, but over time, many smaller dams became technologically obsolete (Walter & Merritts, 2008; Juracek, 2015). Dams may dramatically alter stream physicochemical parameters, hydrological processes, aquatic communities and ecosystem function (reviewed in Baxter, 1977; Ligon et al., 1995; Lessard & Hayes, 2003; Graf, 2006). Primary impacts of dams include restricted organism movement, alteration of stream habitat, reduced sediment movement and alteration of water chemistry parameters including temperature, DO, and conductivity. All of these changes likely have strong negative consequences for stream organisms (Baxter, 1977; Poff et al., 1997; Beneteau et al., 2009).

Dams primarily alter stream hydrology and physicochemical habitat parameters by reducing the extremes of high and low water events and by creating impoundments. The magnitude and frequency of high and low flows regulate many ecological and physical processes including species composition and downstream sediment dispersal (Poff et al., 1997). Moderated flow regimes may negatively impact migratory fishes by reducing migration cues associated with long flood pulses in rivers with both small and large dams (Osmundson et al., 2002; Lytle & Poff, 2004; Zigler et al., 2004; Gubiani et al., 2007). Small, surface-release dams may elevate downstream water temperatures which can negatively impact many fishes and other sensitive aquatic taxa. Because warmer water holds less DO than does cooler water, reaches downstream of some dams may be oxygen-depleted. Additionally, thermal life-history cues may be dampened

or eliminated (Irvine, 1985; Lytle & Poff, 2004). Coarsening of the substrate due to scouring downstream of small dams may also reduce available habitat for benthic organisms, resulting in lower diversity (Hauer et al., 1989; Beasley & Hightower, 2000). Moreover, increased sediment retention in impoundments and in de-watered tailwaters may eliminate sediment-intolerant taxa (Osmundson et al., 2002; Bessert & Orti, 2008). These negative effects caused by the presence of a dam have the potential to shift population dynamics and directly alter ecosystem processes that are vital to aquatic systems.

Dams can alter stream physicochemical parameters including water temperatures, levels of DO and sedimentation (Fairchild & Velinsky, 2006; Gangloff et al., 2011; Hoch, 2012; Holcomb, 2013). Dams inundate upstream free-flowing reaches and form reservoirs where flow levels are reduced. Impoundment habitats are often nutrient and sediment sinks. Nutrient sinks promote eutrophication and unfavorable conditions for many aquatic organisms vital to the integrity of the stream (Anderson et al., 2002). More recent studies suggest that some low-head structures may provide some ecological benefit to freshwater biota. Some species of freshwater mussels appear to benefit from small dams (Singer & Gangloff, 2011). Gangloff et al. (2011) found that intact low-head dams have a positive effect on freshwater mussel abundance and diversity, and that breached dams appear to have strong negative effects on mussel populations. These positive effects on mussels may be mediated by substrate, water temperature, and seston export and quality. Additionally, Helms et al. (2009) reported higher fish assemblage diversity immediately downstream of breached low-head dams compared to upstream sites in Alabama and that substrate was the greatest predictor of fish taxon and trait richness. These findings are somewhat counter-intuitive as they directly contrast many widely-held presumptions about dam effects on stream biota.

Dam removal is an important component of many stream restoration projects; however, little is known about how the ecological and physical changes that follow dam removal affect stream biota. Insect and fish abundance can increase following the breach and removal of low-head dams (Maloney et al., 2008; Pess et al., 2008; Brainwood et al., 2008); however, sediments that have accumulated behind the dam are released downstream and may be contaminated with PCB's, agricultural nutrients, and other pollutants (Stanley et al. 2002; Doyle et al., 2003). Sediments in formerly impounded reaches are relatively unstable and channel re-development is typically slow (Stanley et al., 2002; Doyle et al., 2003). A low-head dam removed in the Hudson River released PCB contaminated sediments and, within one year, striped bass (*Morone saxatilis*) in downstream reaches had significantly elevated tissue PCB levels (Barnthouse et al., 2003). Although my study suggests the impact of downstream sediment release on biota, few studies have examined the effects of dam removal on stream ecosystem functions.

Aquatic macroinvertebrates are often used as biotic indicators because many species are intolerant of degraded habitat or water quality (Rehn, 2008). In areas where a dam is present, macroinvertebrate abundance may be altered due to changes in stream habitat conditions, increased temperatures, altered nutrients, and lowered DO (Carlisle et al., 2014). It is important to understand the effects of dams on macroinvertebrate communities because macroinvertebrates have important functional roles in stream ecosystems, including being major drivers of leaf decomposition (Cummins et al., 1989; Gessner & Chauvet, 2002; Rehn, 2008). Benthic invertebrates are classified into functional feeding groups (Wallace & Webster, 1996). Grazers and scrapers feed on biofilm that accumulates from colonizing microorganisms on the surface of leaves and rocks. Collectors and filterers feed on fine particulate organic matter (FPOM) taken directly from substrate (collector) or filtered from the water column (filterers). Shredders feed on

coarse particulate organic matter (CPOM) such as leaves and other detritus (Merritt & Cummins, 1996). Stream detritivores (i.e., shredders) play a pivotal role in the flow of nutrients within stream ecosystems. Leaf fragments and fecal matter produced by shredders results in the production of FPOM which is utilized by other aquatic organisms (Petersen & Cummins, 1974; Wallace et al., 1995). Either the presence of a dam or disturbance from its removal could negatively influence aquatic insect communities and alter the entire dynamic of a stream.

Shredders may have a remarkable impact on the rates of leaf decomposition and FPOM production (Graca et al., 2001; Lecerf et al., 2006). Wallace et al. (1982) treated a first order Appalachian stream with insecticide and reduced aquatic insect abundance to <10% of pretreated levels. When compared to an untreated study stream, the removal of invertebrates resulted in a 70% reduction of leaf decomposition and 30% reduction in FPOM production. Because macroinvertebrates frequently drive leaf breakdown rates in many temperate streams, it is likely that where dams change invertebrate communities they also indirectly alter decomposition and nutrient release rates (Vannote et al., 1980; Rehn, 2008).

With the exception of some crayfishes, most shredders are unable to digest leaf material prior to accumulation of biofilm (Gartner & Cardon, 2004). Leaves entering low order streams are immediately colonized by microorganisms that condition leaf material and make it more palitable to invertebrate consumers (Suberkropp, 1992a; Gessner & Chauvet, 1994; Weyers & Suberkropp, 1996). Colonization by microbes initiates leaf degradation, making leaf tissue more accessible to the shredders and increases the flow of nutrients within a system. During senescence, nutrients are reabsorbed by plants, decreasing the nutrient content within the leaves (Stapel & Hemminga, 1997). Fungal mycelia create a biofilm on leaves elevating the nitrogen content and adding substantial nutritional value (Suberkropp & Klug, 1976; Chauvet, 1987).

Although microbial colonization starts prior to immersion, microbial biomass associated with leaves is greatly increased once leaves enter aquatic systems (Gessner & Chauvet, 1997; Gulis & Suberkropp, 2003). Water chemistry, temperature and the structural composition of leaves all play a role in determining the development and activity of microbes (Suberkropp & Chauvet, 1995; Chung & Suberkropp, 2008).

Extracellular enzymes associated with fungi and bacteria precede invertebrate colonization and in addition to facilitating shredder feeding, microbial conditioning may also be a determinant of leaf mass loss (Hieber & Gessner, 2002; Suberkropp, 1992a; Suberkropp, 1992b; Imberger et al., 2008). Microorganisms increase aquatic leaf breakdown directly by releasing cellulase, pectinase and other enzymes (Findlay & Arsuffi, 1989; Gessner & Chauvet, 2002; Suberkropp, 1998). Numerous studies have shown that microbial decomposition may, in some streams, exceed invertebrate-mediated breakdown (Imberger et al., 2008; Simon et al. 2009; Robinson et al., 1998). Moreover, in systems where detritivorous macroinvertebrates are uncommon or absent (e.g., impoundments and many natural lakes) microbial decomposition may be the primary pathway for CPOM breakdown, FPOM production and nutrient mobilization (Cummins & Klug,1979; Gessner & Chauvet, 1994; Imberger et al., 2008). Microorganisms are therefore essential to biologically-mediated decomposition, but their role as modifiers should not overshadow their direct influence on leaf breakdown rates. More emphasis is now being placed on understanding the direct role of microbial communities in leaf decomposition in aquatic ecosystems (Hieber & Gessner, 2002).

This study documents two experiments showing the effects of altered physicochemical conditions on the rate of leaf decomposition up- and downstream from a small dam. My main objectives were to: (1) document how changes in temperature, conductivity, and DO associated

with an impoundment affects the rate of decomposition, (2) evaluate the possibility that reaches upstream and downstream of dams may differ in the rate of decomposition as well as dam status, and (3) determine whether macroinvertebrate shredding or microbial colonization drives leaf decomposition rates. I hypothesize that due to altered physicochemical conditions that may deter aquatic macroinvertebrate colonization, decomposition will be significantly lower in impoundments compared to nearby free-flowing reaches. Further, I hypothesize that downstream from dams, elevated nutrient concentrations will increase microbial colonization, invertebrate-mediated leaf processing, and overall decomposition rates.

Methods

Experiment 1- Dam Status and Leaf Decomposition

Study Sites

I conducted the first experiment within two high-gradient fourth-order streams in the New and Watauga river basins in northwestern North Carolina. The mean annual temperature of this region is 16.0˚C, with a mean annual precipitation of 133.7 cm (NOAA—Southeast Regional Climate Center). I chose the three sites because they are located on similarly-sized streams but have dams with differing conditions. Shull's Mill (36° 10' 51.7692" N 81° 43' 30.4572" W) is a breached dam located on the upper Watauga River near the historical settlement of Shulls Mill, with operations beginning around 1917 and ending in 1940 following a large flood. Today much of the dam remains in the channel however the channel has migrated around the structure. Ward's Mill (32° 58' 17.4468" N 85° 20' 37.8096" W) is an intact dam located on the upper Watauga River near the town of Valle Crucis, North Carolina. It was built in 1890 by Bill Ward and was originally powered by a Pelton Wheel. Several flooding events throughout the years have damaged the mill however the mill still remains intact today. After the flood of 1940, much of the dam was in ruins and was shut down in 1947. However, the dam was re-built by the Ward Family in the latter part of the 20th century and currently the dam provides low-wattage hydroelectric power. The New River Light and Power Dam (36° 11' 53.4480" N 81° 39' 59.7924") is a relict dam that began its operation in 1915, generating electricity for Appalachian Training School (now Appalachian State University) and the town of Boone, North Carolina. In 1923, machinery within the power plant was destroyed by a fire. The dam remained intact until the flood of 1940 which destroyed much of the dam; presently only timbers remain on the streambed.

The entire experimental reach for each dam was ~2000 m with four sub-reaches located in 1) an upstream free-flowing reach, 2) within the impoundment, 3) a reach located just downstream of the dam or former dam site (tailwater reach) and 4) a downstream reference reach located >500 m from the dam or former dam site (Fig. 1). An example of this experimental design with corresponding sub reaches can be see at my intact site, Wards Dam (Fig. 2). Sub-reaches were located in sections with similar substrate, depth and flow conditions although habitat in the Ward's Mill impoundment is characterized by very low flows.

Leaf Decomposition

I collected sugar maple (*Acer saccrum*) leaves from trees in Boone, NC, just prior to abscission. I dried leaves in brown paper bags overnight at 50° C in a drying oven. I weighed dried leaves , organized them into 4.0 ± 0.6 g batches, and placed them in plastic 3.8 L containers filled with DI water. I allowed leaves to soak for 8 h then clipped them together using 51 mm plastic binder clips. I deployed leaf packs during October of 2012 by attaching binder clips with cable ties to bricks and placing them along the streambed in habitats where leaves would naturally accumulate (e.g., along channel margins). I attached two leaf packs to each brick for a total of 12 leaf packs per site. To decrease the chance of displacement due to large flow or human disturbance, I secured the bricks to the streambed with rebar.

I randomly selected and removed leaf packs from the stream at 4 and 8-week intervals. I carefully removed leaf packs from bricks, placed them into plastic bags and then transferred them on ice to the lab for processing. First, I placed leaf packs into individual containers to remove accumulated sediments and invertebrates then moved them to paper bags to dry in a

drying oven at 50° C for 24h. Next, I re-weighted the leaf packs to determine mass loss and processing rate coefficients (*k*) (Muto et al., 2011).

I established a leaching control for each site and reach. I deployed the leaching controls as described above but removed them after only 48 h. Because sugar maple leaves lose up to 25% of their mass within the first 24 h, it was critical to include this process to calculate mass loss due to macroinvertebrate activity (Maxted et al., 2005).

Macroinvertebrate Processing

I removed macroinvertebrates from leaf packs in the lab. After leaves were cleaned and transferred to bags for drying, I filtered the water used to rinse each leaf pack through a 500 µm sieve and placed invertebrates into 70% ethanol. I enumerated benthic macroinvertebrates and identified them to family using keys in Merritt and Cummins (2008) and Thorp and Covich (2001). I identified Ephemeroptera, Plecoptera and Trichoptera (EPT) taxa to genus.

Water Quality

I measured water quality parameters (stream temperature, pH, DO (% saturation), and specific conductance) at each site on the day of leaf pack deployment, two weeks into the experiment, and at the time of leaf pack removal using a Yellow Springs Instruments (YSI) professional plus series meter (Yellow Springs Instruments, Yellow Springs, OH)).

Statistical Analyses

I calculated mean reach-scale decomposition (% mass lost) for all streams and reaches. I used an exponential decay model ($M_t = M_0 \times e^{-kt}$) to determine leaf processing coefficients (*k*/day). I

used a 2-way analysis of variance (ANOVA) and least significant difference (LSD) post hoc tests to test the hypothesis that decomposition rates vary among dam types. I used 1-way ANOVAs to test the hypothesis that stream position (up- vs. downstream from dams) had an effect on leaf decomposition, invertebrate communities and water chemistry parameters within each stream. To test for correlations between water chemistry parameters, invertebrate metrics and among-site decomposition rates, I used Pearson product-moment correlation coefficients. I preformed all statistical analyses using IBM SPSS statistical software (Version 20).

I assigned the shredder functional feeding group using Merrit & Cummins (2008). I calculated total mean abundance, EPT richness, total number of shredders, percent shredders, and Shannon-Wiener Diversity index (H') for each leaf pack.

Experiment 2- Microbial Versus Invertebrate Decomposition

Study site

I conducted the second study within the upper Watauga River in the southern Appalachian Mountains of North Carolina. I selected four sites within an ~2 km reach associated with Ward's Mill Dam near the town of Valle Crucis, North Carolina (experiment 1:intact dam). I selected a site in an upstream free-flowing reach ~400-550 m from the intact dam to serve as a control for dam effects, a site within the impoundment located 0-150 m from the dam, a site located just downstream (0-150 m) of the dam (tailwater reach), and a downstream free-flowing reach located ~600-750 m downstream from the dam (Fig 2).

Leaf Decomposition

I collected freshly-senesced sugar maple (*Acer saccrum)* leaves immediately following abscission. I dried leaves in brown paper bags overnight (~12 h) in a drying oven at 50 ˚C. I weighed and organized leaves into into 4.0 ± 0.6 g batches, either clipped with a binder clip or placed in 16 x 20 cm fine-mesh (0.5 mm) bags. I used fine mesh bags to measure microbial decomposition by restricting macroinvertebrate access to the leaf pack. I anchored 9 replicates of each treatment to the streambed using bricks, cable ties and rebar at each site. At the beginning of the study, I randomly selected three leaf packs of each treatment from each site after 30 minutes and then again after 48 h as handling and leaching controls, respectively. The experiment ran for a total of 12 weeks between November 2013 and February 2014. I randomly retrieved 3 replicates of each treatment at 4 week intervals over the course of the study.

I placed leaf packs directly into plastic bags in the field and transferred the packs to the laboratory on ice for processing. I rinsed leaves with deionized water into a 400-µm screen to remove sediment and invertebrates. I preserved macroinvertebrates in 70% ethanol. I dried leaves at 50 ˚C for 48 h and determined leaf decomposition rates (*k*). I measured leaf mass loss (LML) due to leaching and handling and accounted for handling loss in subsequent analyses.

Macroinvertebrate assemblages

I sorted preserved macroinvertebrates under a dissecting microscope. I identified EPT taxa to the lowest possible taxon (usually genus) and enumerated them. I enumerated other benthic macroinvertebrates and identified them to family using keys in Thorp and Covich (2001) or Merritt and Cummins (2008). I assigned EPT taxa to functional feeding groups with statistical analysis focused on leaf-shredding macroinvertebrates. I calculated mean total abundance, EPT

richness, total number of shredders, percent shredders, and Shannon-Wiener Diversity index (H') for each leaf pack.

Crayfish sampling

In addition to aquatic insects, I also used 1 m^2 kick-nets to assess crayfish abundance at each site during the first 4 weeks of the study. I took four 30-second kick-net samples covering in riffles and pool microhabitats at each site.

Habitat and Water Quality

At each study site, I measured depth and flow at five equidistant points using a meter stick and flow meter (Marsh-McBurney Flow-Mate model 2000). I measured several water quality parameters at each site to record physical and chemical changes throughout the experiment. I recorded stream temperature, DO (% saturation), and specific conductance on the day of leaf pack deployment using a YSI Professional Plus series meter (Yellow Springs Instruments, Yellow Springs, OH), 2 weeks into the experiment and at the time of leaf pack removal.

Statistical Anaysis

I assumed that overall decomposition (% mass lost) for each site and treatment to be the difference between the initial dry mass (g) and final dry mass (g) (Muto et al., 2011). I used an exponential decay model (Mt = M0 x e-kt) to determine leaf processing rates (*k*/day) for each site and treatment. I used a 2-way ANOVA with least significant difference (LSD) post hoc tests to test the hypothesis that decomposition rates were similar among all sites and treatments. To examine correlations between water chemistry parameters, invertebrate metrics and

decomposition rates, I used Pearson product-moment correlation coefficients. I preformed all statistical analyses using IBM SPSS statistical software (Version 20).

Results

Experiment 1

Leaf Decomposition

I did not observe among-reach LML differences at any streams. After eight weeks, leaf mass in all leaf packs had declined by >40%. During this interval average stream temperatures dropped from 12 ˚C to 3 ˚C. LML was highest in the impoundment of the breached dam but was also high in the tailwaters of the intact dam (Fig. 3). Although LML differences among reaches in streams with breached and relict dams were not significant, The exponential decay model used to calculate the processing coefficients (*k*/day) for each site revealed the highest processing rates at the relict site and the highest mean LML at the breached site (Table 2). There was a significant effect of stream position (reach) on LML at sites associated with the intact dam. At the intact dam site, tailwater LML was significantly greater than impoundment LML (Fig. 3). Pearson correlations revealed a negative correlation between specific conductance and LML (Fig. 4).

Macroinvertebrate Communities

Across all sites, LML was positively correlated with EPT richness and mean total abundance (Fig. 5, Table 1). Although LML was positively correlated with EPT richness and mean total abundance, there was no significant relationship between LML and shredder abundance suggesting that EPT taxa may not be driving LML in these streams (Fig. 5). Additionally, the

highest rate of leaf decomposition (k = -0.0156 to -0.0212) was seen at the South Fork New River relict dam site where invertebrate and shredder abundance were lowest (Table 1, Table 2).

Water Quality

I recorded mean water quality data for all reaches associated with each dam site (Table 2). Although increased temperatures were associated with decreased LML, the relationship was not statistically significant. Specific conductance was significantly negatively correlated with site-scale 8-week LML (Fig. 4, Table 2).

Experiment 2

Leaf Decomposition

At the end of the second experiment, leaf mass in all leaf packs declined by >50%. LML was significantly greater at both 8 and 12 weeks when compared to the 4 week trials, with the highest total loss in leaf mass occurring between 8 and 12 weeks (Fig. 6). There was a significant difference in decomposition between treatments among sites. When both microbial communities and aquatic insects had access to the leaf packs, LML differences increased significantly (Fig. 7). After 12 weeks, leaf packs exposed to microbial conditioning and macroinvertebrate shredding at all sites had higher decomposition rates compared to leaf packs where macroinvertebrates were excluded (Fig. 7). I observed a significant effect of site and treatment on decomposition with highest LML recorded in the tailwaters of exposed leaf packs (Fig. 7). The exponential decay model used to calculate decomposition rates (k/day) for each site and treatment revealed that the rate of decomposition was higher in exposed leaf packs for all sites except for sites located within the impoundment (Table 3).

Macroinvertebrate Communities

Across all sites, there was no significant correlation between macroinvertebrate community metrics and LML after 12 weeks (Fig. 8). Total abundance of macroinvertebrates in the tailwater site was comparable to the upstream control site which had the highest diversity (H') and richness (Table 4). Results of my experiment, although not significant, are biologically important because sensitive aquatic taxa appear to be unaffected by habitat conditions in the tailwater. I measured the highest rate of decomposition (k = -0.0129) in the tailwater reach immediately downstream of the dam. At this site, invertebrate and shredder abundance were higher than at other sites (Table 4). Elevated LML rates in this reach are most likely attributable to accelerated conditioning from nutrient enriched waters and increased shredder abundance and fragmentation.

No crayfish were detected in any of the microhabitats within each site and other recent surveys of the Watauga River have demonstrated that crayfishes are uncommon in this reach (Gangloff et al. unpublished data).

Water Quality

I measured and recorded water quality data for all sites (Table 3). There was no significant effect of water quality parameters measured on LML (Fig. 9). There was an observable association between flow (m/s) and LML which is likely biologically significant (Fig. 10).

Discussion

My study demonstrates that small dams appear to have counter-intuitive effects on stream physicochemical parameters, biota and ecosystem processes. These changes were most evident in sub-reaches associated with an intact dam (Wards Dam). At this intact dam site, I observed significantly higher LML in the tailwater compared to the impoundment. However, I also observed higher decomposition rates in dam tailwaters compared to sites located ~500 m up- and down-stream from the impoundment. Although these differences were not statistically significant (at the $P = 0.05$ level), there was a notable increase in LML in the tailwaters compared to the up and downstream control sites. This increase in LML in the tailwaters compared to the control sites is likely biologically meaningful; I hypothesize that increased nutrient export from the impoundment site may be increasing microbial colonization of leaf packs and increasing LML in the tailwaters of an intact dam site. Future research should address the role of impoundments in promoting microbial decomposition in tailwaters and downstream reaches.

Stream conductivity and temperature negatively influence decomposition rates by affecting microbial and macroinvertebrate activity. Lower levels of specific conductance and temperature are typically associated with decreased LML (Irons et al., 1994; Robinson et al., 1998). Elevated conductivity and temperature most likely a result of runoff from urbanized tributaries draining the towns of Boone and Blowing Rock. Elevated specific conductance may also be associated with decreased macroinvertebrate abundance or the relative abundance of shredder taxa. However, despite low EPT richness and shredder abundance, the relict site had the highest rate (k) of leaf decomposition again suggesting that microbial communities may be playing an important role in leaf breakdown in urbanized streams (Table 1, Table 2).

Other studies suggest that stream conditions and habitat quality improve following the removal of dams (Bednarek, 2001; Thomas et al., 2005). Interestingly, the New River Light and Power Dam (relict site) is almost entirely gone from the channel and yet I observed elevated conductivity levels and the lowest overall LML. These findings suggest that this structure may still be affecting stream habitats nearly 75 y post-removal. As conservationists continue to petition the removal of dams, it is imperative that researchers focus more on understanding the potentially beneficial effects that some small dams may have on the surrounding ecosystem (Velinsky et al., 2006). Although many small dams are obsolete or deteriorated and may no longer serve their initial purpose, my study suggests that some intact dams may positively influence community structure and stream ecosystem function.

Results from the second study suggest that, in the presence of a small dam, nutrient levels in the impounded site increase microbial colonization and biofilm on leaves. Microbial communities may then become the main driver of decomposition within the impoundment. Biofilm is the main food source for macroinvertebrate communities and where they are able to gain access to leaf packs (exposed) the rate of leaf breakdown increased significantly. Leaf decomposition occurs in a progressive order of three distinct phases: leaching, microbial conditioning and animal shredding (Petersen & Cummins, 1974). Leaching takes place within the first 4 days of leaf entrance into a stream and can lead to substantial leaf mass loss (up to 30%) from withdrawal of soluble material within the leaf (Graca et al., 2001). However, I did not observe significant LML until 8 weeks with the highest loss in leaf tissue between 8 and 12 weeks. After the initial 4 weeks, microbes have had the chance to condition and prime the leaves for future shredders, which is most likely why I saw significantly higher LML between 8 and 12 weeks. At 4 weeks it can be assumed that most LML was due to leaching and initial microbial

decomposition. At 12 weeks, the leaf material had been altered by bacteria and fungi then followed by heavy fragmentation resulting in a total mass loss of >50%. Preferential feeding by macroinvertebrates significantly accelerates leaf mass loss rates over time.

Stream restoration increasingly targets restoring hydrological connectivity and ecosystem function via dam removal even though dam impacts on many stream ecosystem functions remain poorly understood. In the presence of a dam or remnants of prior impoundments, stream physicochemistry may be altered due to altered duration and timing of flow, water temperatures and changes in DO concentrations (Gibson et al., 2005). Within larger impoundments increased sedimentation and accumulation of organic debris may alter nutrient dynamics and export to downstream ecosystems (Baxter, 1977). However, small run-of the river impoundments may not be as retentive as larger impoundments and may even play a role in transforming/mobilizing nutrients (Fairchild & Velinsky, 2006). Because many impoundments are characterized by decreased flow and DO concentrations (conditions unsuitable for many sensitive macroinvertebrate taxa) nutrient mobilization and organic matter transformation is presumably mediated by microbial processes (Rehn, 2008). Because decomposition rates are frequently correlated with stream physicochemistry, abiotic factors should be measured when quantifying ecosystem processes (Velinsky et al., 2006). Elevated temperature and conductivity levels associated with poor water quality could ultimately accelerate microbial growth and increase the rate of leaf decomposition resulting in an overall skewed perception of stream health and integrity.

Impoundments upstream from dams are habitats with reduced flows that can act as nutrient sinks and often limit the abundance of aquatic organisms both within the impoundment and downstream reaches (Fairchild & Velinsky, 2006; Bredenhand & Samways, 2009). My

study suggests that impacts of small dams may be limited to the impounded reach. Further, LML data revealed the highest rate of decomposition (k = -0.0129) in exposed leaf packs of the tailwater reach. Limitations such as these are biologically significant because the impoundment does not seem to be affecting sites immediately downstream of the dam. In most cases, large dams have been shown to alter stream habitats and biota over considerable distances (>10 km) downstream from an impoundment. However my study suggests that some effects may be limited to the impoundements of small dams. For example, there was a complete rebound of macroinvertebrate taxa in the tailwater with dominant genera being considered intolerant (NCBI$\leq$ 3.0). Except within the impoundment, LML was driven by both microbial decomposition and shredders. When macroinvertebrates had access to the leaves LML increased significantly. However, due to habitat conditions within an impoundment, many macroinvertebrate taxa are excluded and therefore microbial communities are the main drivers of decomposition for impounded sites.

Although past research has largely focused on the negative impacts of large dams on stream biota and habitats, little is still known as to how smaller dams impact aquatic taxa, habitats and ecosystem functions. Researchers often focus on macroinvertebrate abundance and species richness but rarely look at how altering the stream could influence trophic dynamics and the interactions between microorganisms and aquatic insects. In the presence of a dam nutrient levels of an impounded site could potentially increase the colonization of microbes and thus the accumulation of biofilm on leaves. Microbes then become the main driver of decomposition within impoundments as most intolerant macroinvertebrate taxa will be excluded.

My study suggests that although the rate of leaf decomposition increased in dam tailwaters, it is most likely attributable to microbial colonization, fragmentation of leaves by

shredders as well as increased flow and abrasion in these high-energy reaches. This study is important to understanding the impact of dams on stream processes and how the presence of a dam could possibly alter the energy dynamics of an impacted system. Dam removal prioritization rubrics attempt to minimize ecological harm while maximizing benefit. However, stream responses to dams are often counter-intuitive and as my study shows, highly site-specific. Dams vary in mode of operation, construction and age which can all play a role in how a system is affected. Future work should examine links between these factors and the localized effects of impoundments on downstream ecosystems. Understanding how ecosystem processes are changed by dams across broader spatial and temporal scales is needed to help conservation stakeholders develop criteria to better evaluate effects of dams on streams prior to removal.

Management Implications & Reccomendations

Although it is well-documented that most dams generally degrade ecosystem structure and function, my research shows more studies need to be conducted to take into consideration the wide range of variation among dam types. Dams range in size, mode of operation, age, and construction which can all play a fundamental role in assessing the potential for restoration (Poff & Hart, 2002). Currently, the standards used for prioritizing dams for removal are largely based on studies of larger structures. More often than not dams are removed with no prior research or follow-up monitoring focused on how removals impact stream ecosystems. Moreover, few studies have considered the effects of dams on stream ecosystem functions. My data suggest that environmental agencies collect more baseline data on the influence of dams on ecological

function as well as on less mobile invertebrate communities prior to and following dam removals (Gangloff, 2013). Although re-establishing connectivity in fragmented river systems is a primary focus for stream conservation projects, it is also important to realize that many systems will be subject to long-term fragmentation by large hydro-electric dams and so benefits to migratory fishes from removing smaller dams may be minimal and countered by the increased risk of exotic fish invasions. My data suggest that it is important to conduct more holistic pre- and post-removal/restoration assessments to better understand impacts of smaller dams in watershed networks. Reconnecting riverine systems through the removal of dams is an ecologically important goal, however it is important to also realize that some dams may apparently enhance ecosystem services as well as protect sensitive biota in stream ecosystems.

References

Anderson, D. M., P. M. Glibert & J. M. Burkholder, 2002. Harmful algal blooms and eutrophication: nutrient sources, composition and consequences. Estuaries 25: 562-584.

Barnthouse, L. W., D. Glaser & J. Young, 2003. Effects of Historic PCB Exposures on the Reproductive Success of the Hudson River Striped Bass Population. Environmental Science and Technology 37: 223-228.

Baxter, R. M., 1977. Environmental effects of dams and impoundments. Annual Review of Ecology and Systematics 8: 255-283.

Beasley, C. A. & J. E. Hightower, 2000. Effects of a low-head dam on the distribution and characteristics of spawning habitat used by striped bass and American shad. Transactions of the American Fisheries Society 129: 1316-1330.

Bednarek, A. T., 2001. Undamming rivers: a review of the ecological impacts of dam removal. Environmental Management 27: 803–814.

Beneteau, C. L., N. E. Mandrak & D. D. Heath, 2009. The effects of river barriers and range expansion of the population genetic structure and stability in Greenside Darter (*Etheostoma blennioides*) populations. Conservative Genetics 10: 477-487.

Bessert, M. L. & G. Orti, 2008. Genetic effects of habitat fragmentation on blue sucker populations in the upper Missouri River (*Cycleptus elongates*). Conservation Genetics 9: 821-832.

Brainwood, M., S. Burgin & M. Byrne, 2008. The impact of small and large impoundments on freshwater mussel distribution in the Hawkesbury-Nepean River, Southeastern Australia. River Research and Applications 24: 1325-1342.

Bredenhand, E. & M. J. Samways, 2009. Impact of a dam on benthic macroinvertebrates in a small river in a biodiversity hotspot: Cape Floristic Region, South Africa. Journal of Insect Conservation 1: 297-307

Carlisle, D. M., S. M. Nelson & K. Eng, 2014. Macroinvertebrate community condition associated with the severity of streamflow alteration. River Research and Applications 30: 29-39.

Chauvet, E., 1987. Changes in the chemical composition of alder, poplar and willow leaves during decomposition in a river. Hydrobiologia 148: 35–44.

Chung, N., & K. Suberkropp, 2008. Influence of shredder feeding and nutrients on fungal activity and community structure in headwater streams. Fundamental and Applied Limnology 173: 35–46.

Cummins, K. W. & M. J. Klug, 1979. Feeding ecology of stream invertebrates. Annual Review of Ecology and Systematics 10: 147-172.

Cummins, K. W., M. A. Wilzbach, D. M. Gates, J. B. Perry & W. B. Taliaferro, 1989. Shredders and riparian vegetation. BioScience 39: 24–30.

Doyle M. W., E. H. Stanley & J. M. Harbor, 2003. Channel adjustments following two dam removals in Wisconsin. Water Resources Bulletin 39: 1011-1026.

Fairchild, W. G. & D. J. Velinsky, 2006. Effects of small ponds on stream water chemistry. Lake and Reservoir Management 22: 321-330.

Findlay, S. E. G. & T. L. Arsuffi, 1989. Microbial growth and detritus transformations during breakdown of leaf litter in a stream. Freshwater Biology 21: 261-269.

Gangloff, M. M., E. E. Hartfield, D. C. Werneke & J. W. Feminalla, 2011. Associations between small dams and mollusk assemblages in Alabama streams. Journal of the North American Benthological Society 30: 1107-1116.

Gangloff, M. M., 2013. Taxonomic and ecological tradeoffs associated with small dam removals. Aquatic Conservation Marine and Freshwater Ecosystems 23: 475-480.

Gartner, T. B. & Z. G. Cardon, 2004. Decomposition dynamics in mixed-species leaf litter. Oikos 104: 230–246.

Gessner, M. O. & E. Chauvet, 1994. Importance of stream microfungi in controlling breakdown rates of leaf litter. Ecology 75: 1807–1817.

Gessner, M. O. & E. Chauvet, 1997. Growth and production of aquatic hyphomycetes in decomposing leaf litter. Limnology and Oceanography 42: 496–505.

Gessner, M. O. & E. Chauvet, 2002. A case for using litter breakdown to assess functional stream integrity. Ecological Applications 12: 498–510.

Gibson, C. A., J. L. Meyer, N. L. Poff, L. E. Hay & A. Georgakakos, 2005. Flow regime alterations under changing climate in two river basins: implications for freshwater ecosystems. River Research and Applications 21: 849-864.

Graca, M. A. S., R. C. F. Ferreira & C. N. Coimbra, 2001. Litter processing along a stream gradient: the role of invertebrates and decomposers. Journal of the North American Benthological Society 20: 408-420.

Graf, W. L., 2006. Downstream hydrologic and geomorphic effects of large dams on American rivers. Geomorphology 79: 336-360.

Gubiani E. A., L. C. Gomes, A. A. Agostinho & E. K. Okada, 2007. Persistence of fish populations in the upper Parana River: effects of water regulation by dams. Ecology of Freshwater Fish 16: 191-197.

Gulis, V., & K. Suberkropp, 2003. Leaf litter decomposition and microbial activity in nutrient-enriched and unaltered reaches of a headwater stream. Freshwater Biology 48: 123–134.

Hauer, F. R., J. A. Stanford & J. V. Ward, 1989. Serial discontinuities in a Rocky Mountain river. II. Distribution and abundance of trichoptera. Regulated Rivers: Research and Management 3: 177-182.

Helms, B. S., J. E. Schoonover & J. W. Feminella, 2009. Assessing influences of hydrology, physicochemistry, and habitat on stream fish assemblages across a changing landscape. Journal of the American Water Resources Association 45: 157-169

Hieber, M. & M. O. Gessner, 2002. Contribution of stream detrivores, fungi, and bacteria to leaf breakdown based on biomass estimates. Ecology 83: 1026–1038.

Hoch, R. A., 2012. Beaver and mill dams alter freshwater mussel habitat, growth and survival in North Carolina piedmont streams. Master's Thesis. Appalachian State University, Boone, NC, USA.

Holcomb, J. M., 2013. The effects of low-head dams and land use change on North Carolina atlantic slope fish community structure. Master's Thesis. Appalachian State University, Boone, NC, USA.

Imberger, S. J., C. J. Walsh & M. R. Grace, 2008. More microbial activity, not abrasive flow or shredder abundance, accelerates breakdown of labile leaf litter in urban streams. Journal of the North American Benthological Society 27: 549-561.

Irvine, J. R., 1985. Effects of successive flow perturbations on stream invertebrates. Canadian Journal of Fisheries and Aquatic Sciences 42: 1922-1927.

Juracek, K. E., 2015. The aging of America's reservoirs: in-reservoir and downstream physical changes and habitat implications. Journal of the American Water Resources Association 51: 168-184.

IBM Corp. Released 2011. IBM SPSS Statistics for Windows, Version 20.0. Armonk, NY, USA: IBM Corp.

Irons III, J. G., M. W. Oswood, R. J. Stout & C. M. Pringle, 1994. Latitudinal patterns in leaf litter breakdown: is temperature really important? Freshwater Biology 32: 401-411.

Lecerf, A., P. Usseglio-Polatera, J. Charcosset, D. Lambrigot, B. Bracht & E. Chauvet, 2006. Assessment of functional integrity of eutrophic streams using litter breakdown and benthic invertebrates. Archiv fur Hydrobiologie 165: 105-126.

Lessard, J. L. & D. B. Hayes, 2003. Effects of elevated water temperature on fish and macroinvertebrate communities below small dams. River Research and Applications 19: 721-732.

Ligon, F. K., W. E. Dietrich & W. J. Trush, 1995. Downstream ecological effects of dams: a geomorphic perspective. BioScience 45: 183-192.

Lytle, D. A. & N. L. Poff, 2004. Adaptation to natural flow regimes. Trends in Ecology and Evolution 19: 94-100.

Maloney, K. O., H. R. Dodd, S. E. Butler & D. H. Wahl, 2008. Changes in macroinvertebrate and fish assemblages in a medium-sized river following breach of a low-head dam. Freshwater Biology 53: 1055-1068.

Maxted, J. R., C. H. McCreedy & M. R. Scarsbrook, 2005. Effects of small ponds on stream water quality and macroinvertebrate communities. New Zealand Journal of Marine and Freshwater Research 39: 1069-1084.

Merritt, R. M. & K. W. Cummins (eds), 1996. An introduction to the aquatic insects of North America, 3rd ed. Kendell/Hunt Publishing Company, Dubuque, Iowa.

Merritt, R. M., K. W. Cummins & M. B. Berg (eds), 2008. An introduction to the aquatic insects of North America, 4th ed. Kendall/Hunt Publishing Company, Dubuque, Iowa.

Muto, E. A., D. P. Kreutzweiser & P. K. Sibley, 2011. Over-winter decomposition and associated macroinvertebrate communities of three deciduous leaf species in forest streams on the Canadian Boreal Shield. Hydrobiologia 658: 111-126.

Osmundson, D. B., R. J. Ryel, V. L. Lamarra & J. Pitlick, 2002. Flow-sediment-biota relations: implications for river regulation effects on native fish abundance. Ecological Applications 12: 1719-1739.

Pess, G. R., M. L. McHenry, T. J. Beechie & J. Davies, 2008. Biological impacts of the Elwha River dams and potential salmonid responses to dam removal. Northwest Science 82: 72-90.

Petersen, R. C. & K. W. Cummins, 1974. Leaf processing in a woodland stream. Freshwater Biology 4: 343–368.

Poff, L. N., D. J. Allan, M. B. Bain, J. R. Karr, K. L. Prestegaard, B. D. Richter, R. E. Sparks & J. C. Stromberg, 1997. The natural flow regime. BioScience 47: 769-784.

Poff, N. L. & D. D. Hart, 2002. How dams vary and why it matters for the emerging science of dam removal. BioScience 52: 659-668.

Rehn, A. C., 2008. Benthic macroinvertebrates as indicators of biological condition below hydropower dams on west slope Sierra Nevada streams, California, USA. River Research and Applications 25: 208-228.

Robinson, T. C., M. O. Gessner & J. Ward, 1998. Leaf breakdown and associated macroinvertebrates in alpine glacial streams. Freshwater Biology 40: 215–228.

Simon, K. S., M. A. Simon & E. F. Benfield, 2009. Variation in ecosystem function in Appalachian streams along an acidity gradient. Ecology 19: 1147–1160.

Singer, E. E. & M. M. Gangloff, 2011. Effects of a small dam on freshwater mussel growth in an Alabama (U.S.A.) stream. Freshwater Biology 56: 1904-1915.

Stanley, E. H., M. A. Luebke & D. Marshall, 2002. Short-term changes in channel form and macroinvertebrate communities following low-head dam removal. Journal of the North American Benthological Society 21: 172-187.

Stapel, J. & M. A. Hemminga, 1997. Nutrient resorption from seagrass leaves. Marine Biology 128: 197-206.

Suberkropp, K. & M. J. Klug, 1976. Fungi and bacteria associated with leaves during processing in a woodland stream. Ecology 57: 707-719.

Suberkropp, K., 1992a. Interactions with invertebrates. In: Bärlocher, F (eds) The ecology of aquatic hyphomycetes. Springer, Berlin 118–134.

Suberkropp, K., 1992b. Aquatic hyphomycete communities. In: Wicklow DT, Carroll GC (eds) The fungal community: its organization and role in the ecosystem. Dekker, New York 729–747.

Suberkropp, K. 1992b. Aquatic hyphomycetes communities: the fungal community. Its organization and role in the ecosystem.

Suberkropp, K. & E. Chauvet, 1995. Regulation of leaf breakdown by fungi in streams: influences of water chemistry. Ecology 76: 1433–1445.

Suberkropp, K. & J. B. Wallace, 1992. Aquatic hyphomycetes in insecticide-treated and untreated streams. Journal of the North American Benthological Society 11: 165–171.

Thomson, J. R., D. D. Hart, D. F. Charles, T. L. Nightengale & D. M. Winter, 2005. Effects of removal of a small dam on downstream macroinvertebrate and algal assemblages in a Pennsylvania stream. Journal of the North American Benthological Society 24: 192-207.

Thorp, P. J., A. P. Covich (eds), 2001. Ecology and classification of North American freshwater invertebrates, 2nd ed. Academic Press Publishing Company, San Diego, California.

Vannote, R. L., G. W. Minshall, K. W. Cummins, J. R. Sedell & C. E. Cushing, 1980. The river continuum concept. Canadian Journal of Fisheries and Aquatic Sciences 37: 130-137.

Velinsky, D. J., K. L. Bushaw-Newton, D. A. Kreeger & T. E. Johnson, 2006. Effects of small dam removal on stream chemistry in southeastern Pennsylvania. Journal of the North American Benthological Society 25: 569–582.

Wallace, J. B., J. R. Webster & T. F. Cuffney, 1982. Stream detritus dynamics: regulation by invertebrate consumers. Oecologia 53: 197–200.

Wallace, J. B., M. R. Whiles, S. Eggert, T. F. Cuffney, G. J. Lugthart, S. U. E. Eggert & K. Chung, 1995. Long-term dynamics of coarse particulate organic matter in three Appalachian Mountain streams. Journal of the North American Benthological Society 14: 217-232.

Wallace, J. B. & J. R. Webster, 1996. The role of macroinvertebrates in stream ecosystem function. Annual Reviews Entomology 41: 115-139

Walter, R. C. & D. J. Merritts, 2008. Natural streams and the legacy of water-powered mills. Science 319: 299-304.

Weyers, H. S. & K. Suberkropp, 1996. Fungal and bacterial production during the breakdown of yellow poplar leaves in 2 streams. Journal of the North American Benthological Society 15: 408–420.

Zigler, S. J., M. R. Dewwy, B. C. Knights, A. L. Runstrom & M. T. Steingraeber, 2004. Hyrdrologic and hydraulic factors affecting passage of paddlefish through dams in the Upper Mississippi River. Transactions of the American Fisheries Society 133: 160-172.

Tables and Figures

Table 1: Macroinvertebrate community metrics measured at all sub-reaches within each dam site for experiment 1. Ephemeroptera, Plecoptera, Trichoptera (EPT), mean total abundance (n), EPT richness, H', total number of shredders and % shredders. Shaded genera represent shredders, the dominant taxa within each site.

Table 2: Water quality parameters for all sub-reaches within each dam site for experiment 1. Mean leaf mass loss throughout this study represented by $\overline{X}$ LML and the relative rate of decomposition for each site and reach reprensented by *k.*

Table 3: Habitat and water quality parameters recorded for each reach at Wards Dam during experiment 2. Note the relationship between increasing flow (m/s) and leaf decomposition. The highest flow 0.504 m/s is associated with the tailwater site where I documented the highest rate in LML (k = -0.0129) and LML (2.26 g).

Table 4: Macroinvertebrate community metrics measured at all sites within Wards Dam for experiment 2. Ephemeroptera, Plecoptera, Trichoptera (EPT), total abundance (n), EPT richness, H', total number of shredders and % shredders. Shaded genera represent shredders, the dominant taxa within each site.

Figure 1: Schematic illustrating the experimental design for each site (experiment 1 & 2). The entire experimental reach for each dam was ~2000 m with four sub-reaches located in 1) an upstream free-flowing reach (control), 2) within the impoundment, 3) a tailwater reach 0-150 m downstream of the dam or former dam site and 4) a downstream reference reach located >500 m downstream from the dam or former dam site.

Figure 2: Map of the study site and four sub-reaches at Ward's Dam in the upper Watauga River of Watauga County, North Carolina. 1) Control site located 400-550 m upstream from dam (WARD US), 2) location impounded by Ward's Dam < 2km (WARD IM), 3) a tailwater reach 0-150 m downstream of dam (WARD TR), and a downstream reference reach located > 500 m downstream from the dam (WARD DS) site.

Figure 3: Box plots of LML in three streams after 8 weeks during experiment 1. Each cluster includes the four reaches for a given dam site. (1) breached (2) relict (3) intact. Reach: (US) upstream, (IM) impoundment, (TW) Tailwaters, and (DS) downstream. LML at 4 weeks was not significant and therefore different among reaches at all sites was shown using 8 week data. Differences in LML were significantly higher in the tailwaters compared to the other sub-reaches. No other dam sites exhibited significant among-reach differences in LML after 8 weeks.

Figure 4: Relationship between 8-week LML and mean physicochemical parameters (A. Temperature, B. % DO Saturation, C. Specific Conductance (µS/cm) and D. pH) at reachees

associated with a breached, intact and relict dam during experiment 1.. There was a significant negative correlation between 8-week LML and specific conductance (r = 0.395; *P*=0.021, n= 4). All other parameters were not significant (n.s.).

Figure 5: Pearson correlation data between 8-week LML and invertebrate community metrics (A % Shredders, B Taxa Richness, C) Shannon-Weiner Diversity (H'), and Total Abundance.at sites associated with three dams during experiment 1. There was a significant positive correlation between 8-week LML and total abundance (r = 0.619, *P*=0.0032, n= 4) and taxa richness (r= 0.765, *P*= 0.0038, n= 4).

Figure 6: Box plots of mean LML for each site within Ward's Dam for experiment 2. LML was significantly greater between 8 and 12 weeks (*P*=0.015, F=9.29, df= 2) than 4 and 8 weeks (*P*= 0.018, F= 8.53, df= 2). At the end of the 12 week study leaf packs had declined >50%.

Figure 7: Two-panel box plot figure showing treatment effects on 12-week LML and relative macroinvertebrate abundance. When leaf packs were accessible to microbial communities and invertebrate shredders (Panel A) there was a significant increase in LML over 12 weeks (*P*= 0.005, F= 5.64, df= 2) with the highest decomposition associated with leaf packs in the tailwater reaches. After 12 weeks leaf packs in the open treatment had the highest rate of decomposition

(Panel B). Note that macroinvertebrate abundance was significantly decreased due to treatment, with macroinvertebrate abundance significantly higher in open leaf pack treatments (P= 0.005).

Figure 8: Association between between 12-week LML and macroinvertebrate community metrics (A % Shredders, B Taxa Richness, C Shannon-Weiner Diversity (H') and D Total Invertebrate Abundance) at reaches associated with Wards Mill Dam in the Watauga River for experiment 2: There were no significant correlations between macroinvertebrate metrics and 12-week LML (n.s.).

Figure 9: Relationship between 12-week LML and flow (m/s) for each site. Although not statistically significant (n.s.) there is a notable increase in leaf decomposition where increasing flow rates were recorded.

Figure 10: Association between between 12-week LML and physicochemical parameters (A. Temperature, B. % DO Saturation, C. Specific Conductance (µS/cm) and D. pH) at sites associated with Wards Mill Dam in the Watauga River for experiment 2. There were no significant correlations between physicochemical parameters and 12-week LML (n.s.).

Table 1:

EPT Taxa	NCBI	Breached	Relict	Intact
Heptagenidae				
Maccaffertium	3.2	16	5	22
Epeorus	1.3	1		
Hydropsychidae				
Cheumatopsyche	6.2	9	6	6
Certatopsyche	3.1	2		2
Diplectrona	2.2	1		
Leuctridae				
Leuctra	0.7	296	33	79
Ephemerellidae				
Ephemerella	2.0	5	1	1
Ephemera	1.1	1		
Eurylophella	4.3	2		
Taenipterygidae				
Oemopteryx	1.0	8	2	3
Taeniopteryx	5.0	13	2	7
Polycentropodidae				
Cernotina	4.0	1	3	
Perlodidae				
Malirekus	1.0	1	2	2
Isoperla	2.0	3		1
Diploperla	2.0	1		
Philopotamidae				
Wormaldia	0.7	1		
Chloroperlidae				
Haploperla	1.0	2		1
Isonychiidae				
Isonychia	3.5	1		
Leptophlebidae				
Leptophlebia	6.2	10		1
Paraleptophlebia	6.2	1		
Baeitidae				

Baetis	4.5	5		2
Glossosmatidae				
Glossosoma	1.6			1
Abundance (n)		380	54	130
EPT Richness		21	8	13
Shannon H'		-2.16	-2.58	-2.58
Total Shredder (n)		317	37	89
% Shredders		0.83	0.83	0.68

Table 2:

Site	DO% (µS/cm)	pH	Water Temperature	Specific Conductance	Salinity (‰)	k	x̄ LML (g)
Breached							
US	96.6	8.2	6.8	33.6	0.03	-0.0097	2.24
IM	99.1	8.4	6.8	37.8	0.03	-0.0103	2.16
TW	97.4	8.4	6.9	37.9	0.03	-0.0131	1.82
DS	98.1	8.5	6.9	36.8	0.03	-0.0156	1.57
Relict							
US	103.1	8.1	7.2	90.7	0.07	-0.0212	1.14
IM	108.3	8.0	7.1	90.2	0.07	-0.0187	1.30
TW	100.1	8.0	6.9	90.2	0.07	-0.0156	1.57
DS	97.4	8.1	6.9	90.3	0.07	-0.0173	1.42
Intact							
US	103.9	8.3	6.7	69.7	0.05	-0.0175	1.40
IM	101.5	8.1	6.2	65.0	0.04	-0.0168	1.46
TW	104.1	7.9	5.9	65.7	0.04	-0.0116	2.00
DS	111.9	8.2	6.5	63.9	0.04	-0.0160	1.53

Table 3:

All Sites	DO% (µS/cm)	pH	Water Temp	Specific Conductance	Salinity (%)	Depth (m)	Flow (m/s)	*k*	x̄ LML (g)
Upstream	105.6	7.9	10.5	73.7	0.03	0.655	0.447	-0.0121	2.02
Impoundment	95.3	7.7	8.7	84.3	0.04	1.241	0.211	-0.0075	1.64
Tailwater	101.7	7.3	7.6	82.4	0.04	0.614	0.504*	-0.0129	2.26
Downstream	105.4	7.8	7.3	78.5	0.04	0.478	0.395	-0.0101	2.01

Table 4:

EPT Taxa	NCBI	Upstream	Impound	Tailwater	Downstream
Heptagenidae					
Maccaffertium	3.2	23	5	24	11
Epeorus	1.3	7	1	13	9
Stenacron	4.0	3		4	2
Stenonema	3.5	1		1	2
Hydropsychidae					
Cheumatopsyche	6.2	22	8	25	13
Certatopsyche	3.1	19	7	13	12
Leuctridae					
Leuctra	0.7	29	8	22	12
Siphlonuridae					
Siphlonurus	5.8	3	4		1
Ephemerellidae					
Ephemerella	2.0	12	2	18	10
Ephemera	1.1	10		8	6
Eurylophella	4.3	2	2	5	3
Taenipterygidae					
Oemopteryx	1.0	14	5	25	11
Taeniopteryx	5.0	13	10	14	7
Pteronarcyidae					
Pteronarcys	1.6	3	8	7	6
Polycentropodidae					
Polycentropus	3.5	9	3	6	15
Perlidae					
Acroneuria	1.0	16	6	9	7
Perlodidae					
Malirekus	1.0	6		29	9
Isoperla	2.0	9			
Brachycentropodidae					
Brachycentrus	2.1	7			
Philopotamidae					
Wormaldia	0.7	11		9	5
Chloroperlidae					
Allocapnia	1.22	11	4	16	20
Sweltsa	0.0	27		18	15
Isonychiidae					
Isonychia	3.5	1		1	
Leptophlebidae					
Leptophlebia	6.2	4		6	2
Paraleptophlebia	6.2	2			

Baeitidae					
Baetis	4.5	27	12	42	30
Glossosmatidae					
Glossosoma	1.6	18		29	9
Abundance (n)		347	88	360	235
EPT Richness		27	13	24	26
Shannon H'		-3.07	-2.83	-2.96	-2.99
Total Shredder (n)		56	23	61	30
% Shredders		0.16	0.26	0.17	0.13

Figure 1:

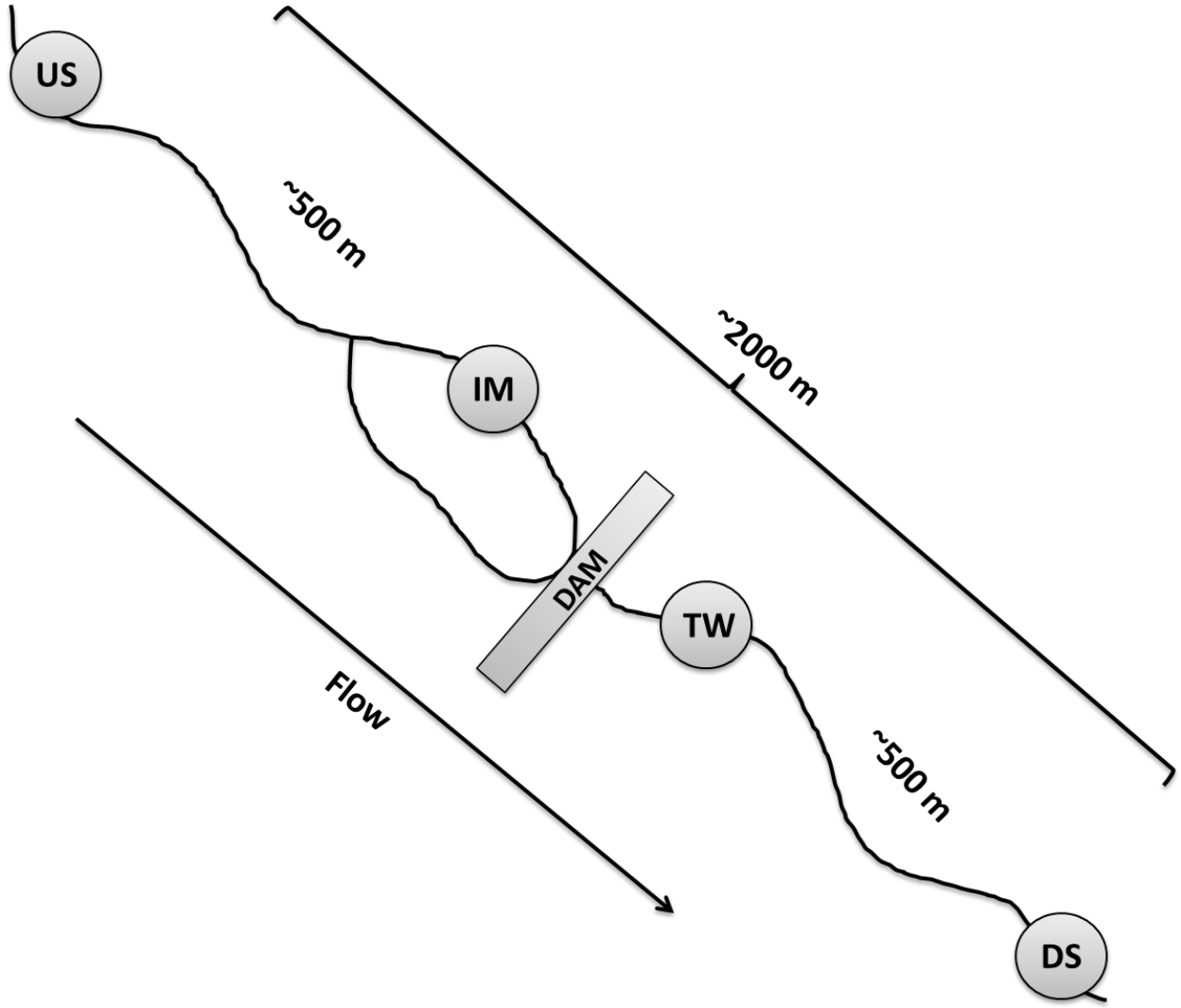


Figure 2:

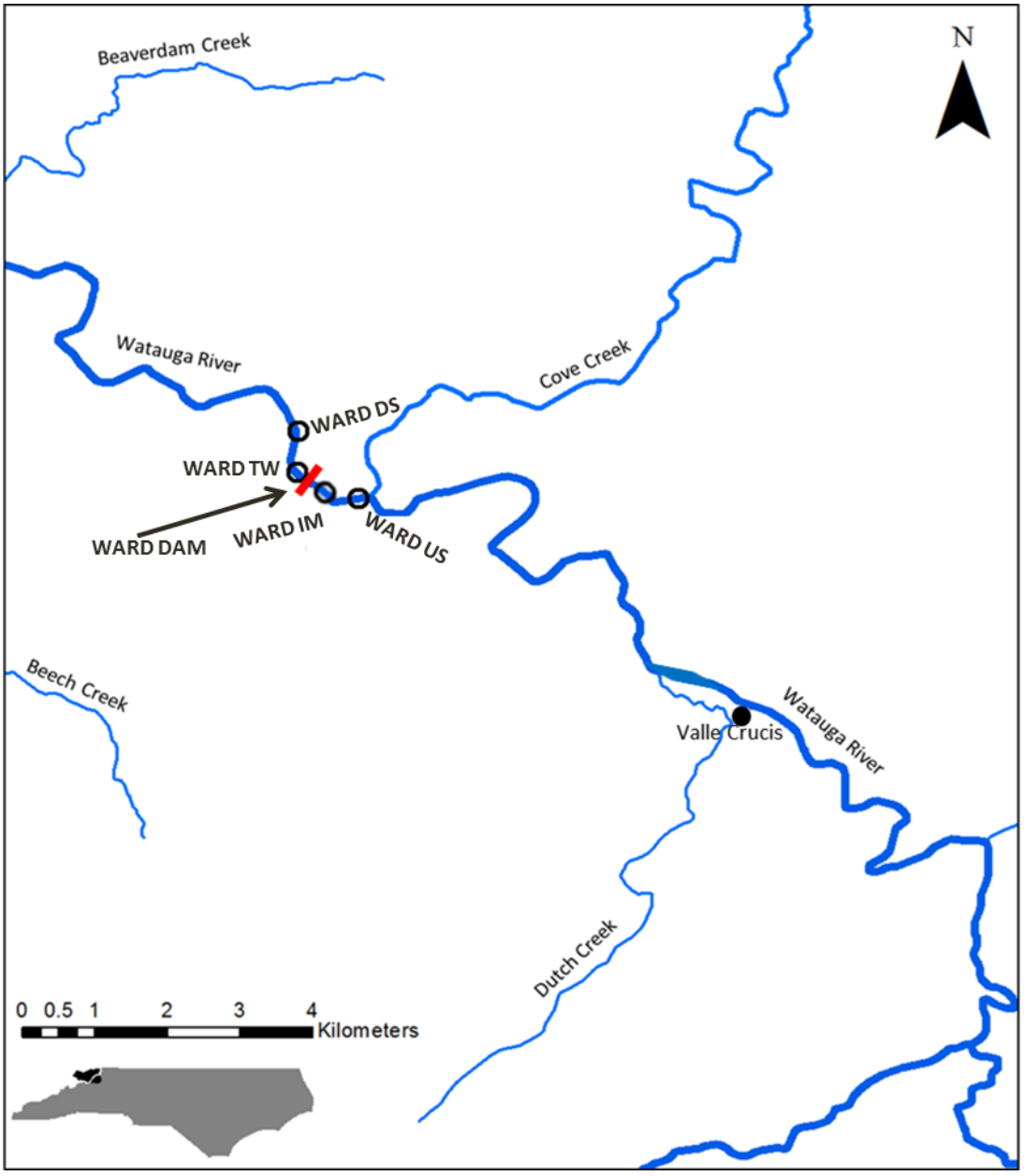

Beaverdam Creek
N
Watauga River
Cove Creek
WARD DS
WARD TW
WARD DAM
WARD IM
WARD US
Beech Creek
Watauga River
Valle Crucis
Dutch Creek
0 0.5 1 2 3 4
Kilometers

Figure 3:

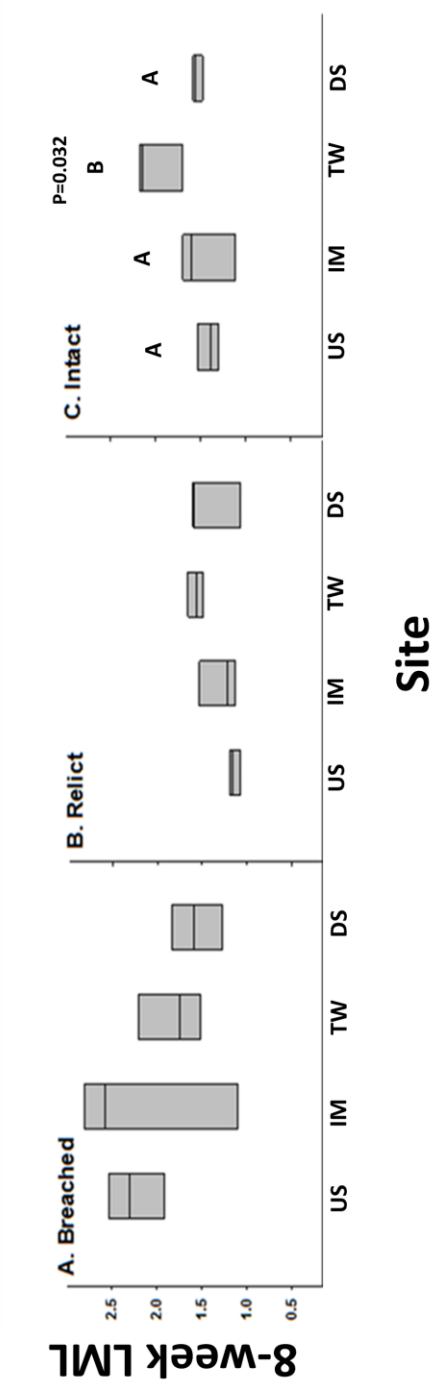

8-week LML
2.5
2.0
1.5
1.0
0.5
A. Breached
B. Relict
C. Intact
P=0.032
A
B
A
A
US
IM
TW
DS
Site

Figure 4:

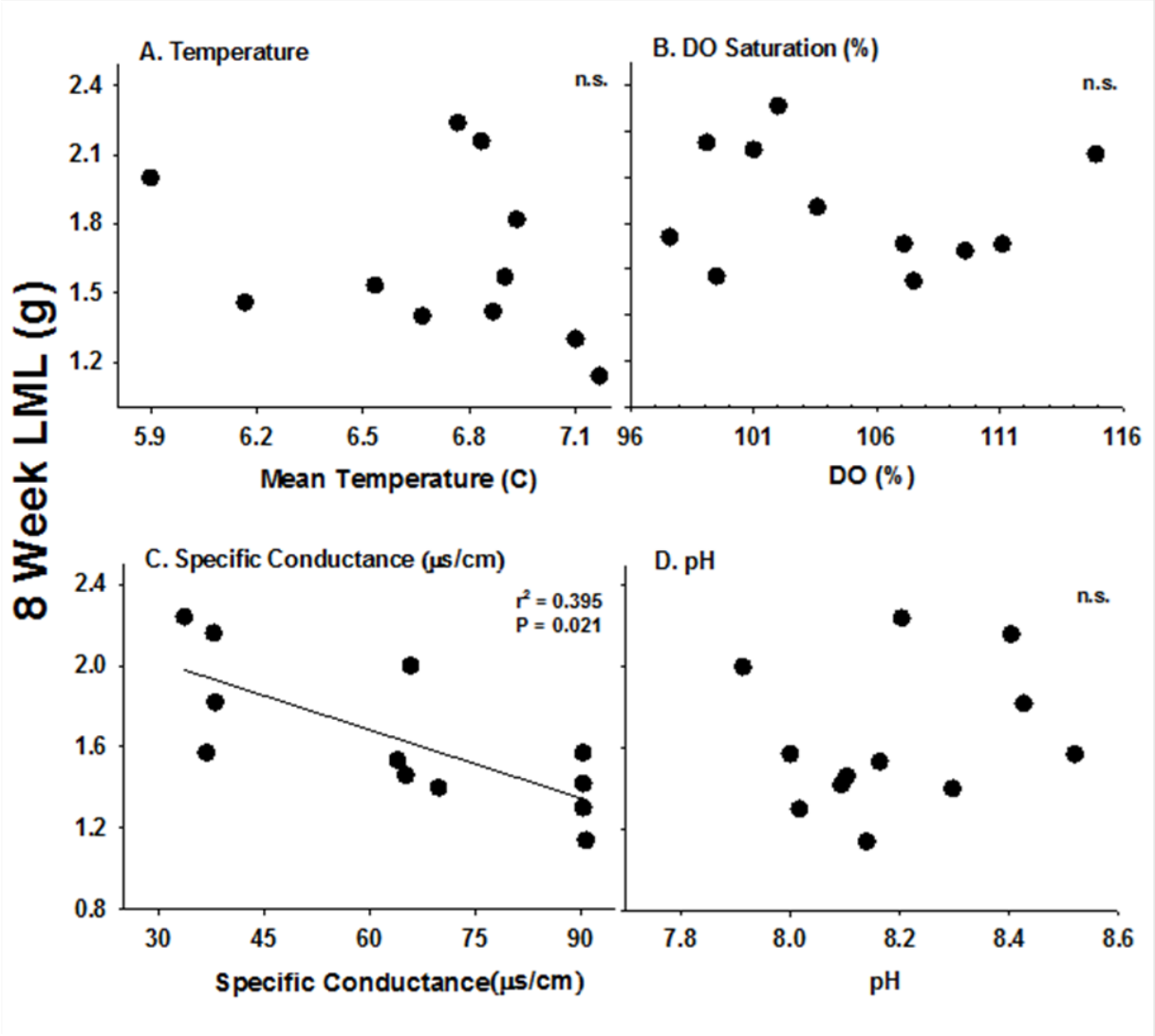

A. Temperature
n.s.
B. DO Saturation (%)
n.s.
8 Week LML (g)
2.4
2.1
1.8
1.5
1.2
5.9
6.2
6.5
6.8
7.1
Mean Temperature (C)
96
101
106
111
116
DO (%)
C. Specific Conductance (µs/cm)
r² = 0.395
P = 0.021
D. pH
n.s.
2.4
2.0
1.6
1.2
0.8
30
45
60
75
90
Specific Conductance(µs/cm)
7.8
8.0
8.2
8.4
8.6
pH

Figure 5:

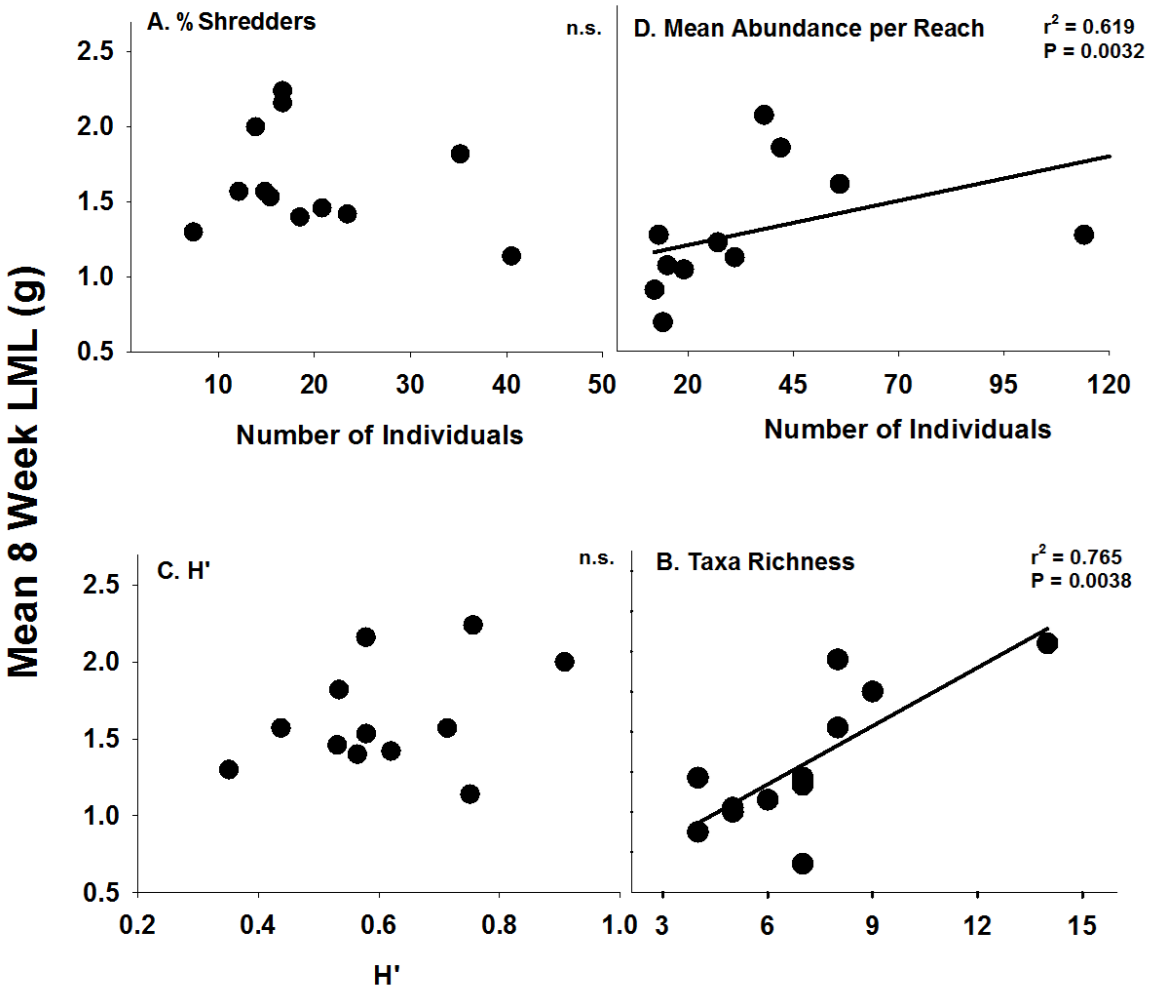

A. % Shredders
n.s.
D. Mean Abundance per Reach
r2 = 0.619
P = 0.0032
Mean 8 Week LML (g)
2.5
2.0
1.5
1.0
0.5
10
20
30
40
50
Number of Individuals
20
45
70
95
120
Number of Individuals
C. H'
n.s.
B. Taxa Richness
r2 = 0.765
P = 0.0038
0.2
0.4
0.6
0.8
1.0
H'
3
6
9
12
15

Figure 6:

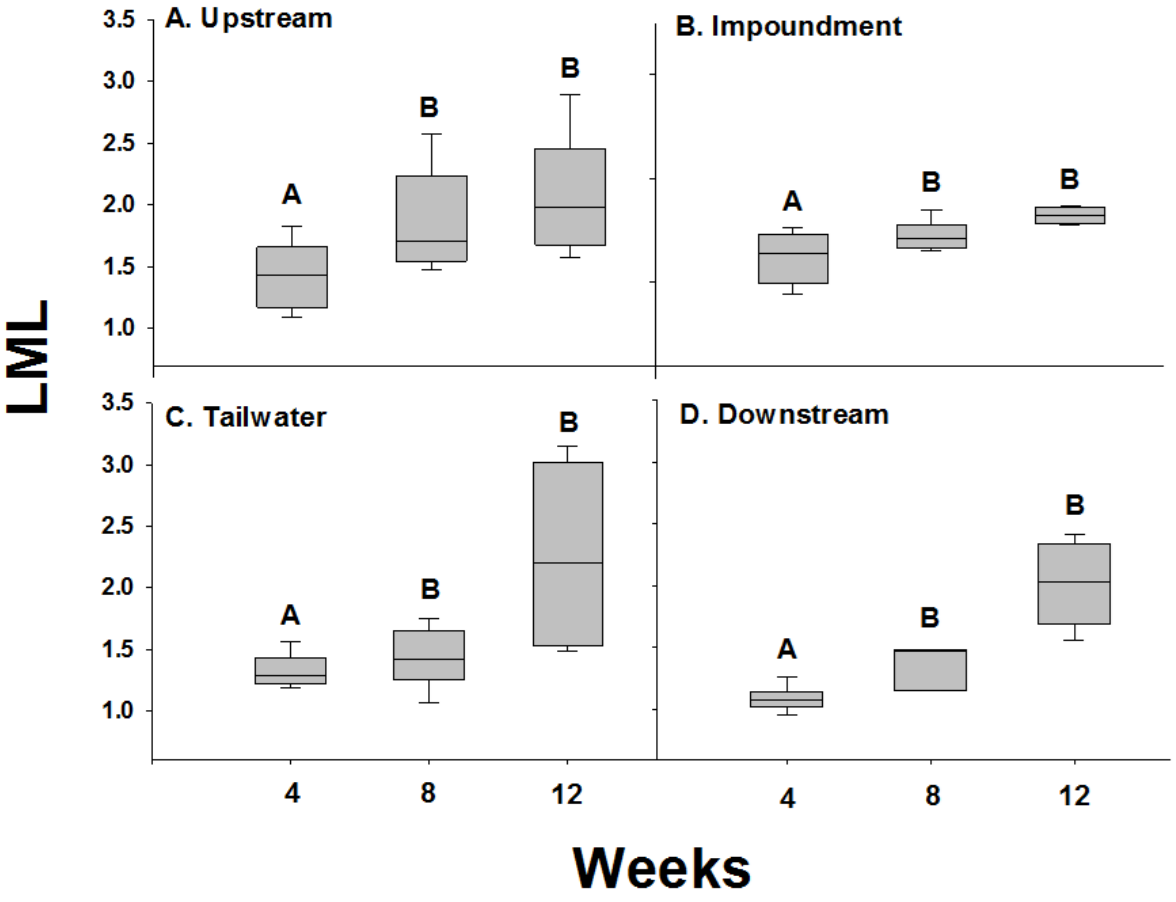

A. Upstream
B. Impoundment
C. Tailwater
D. Downstream
LML
Weeks
3.5
3.0
2.5
2.0
1.5
1.0
4
8
12
A
B

Figure 7:

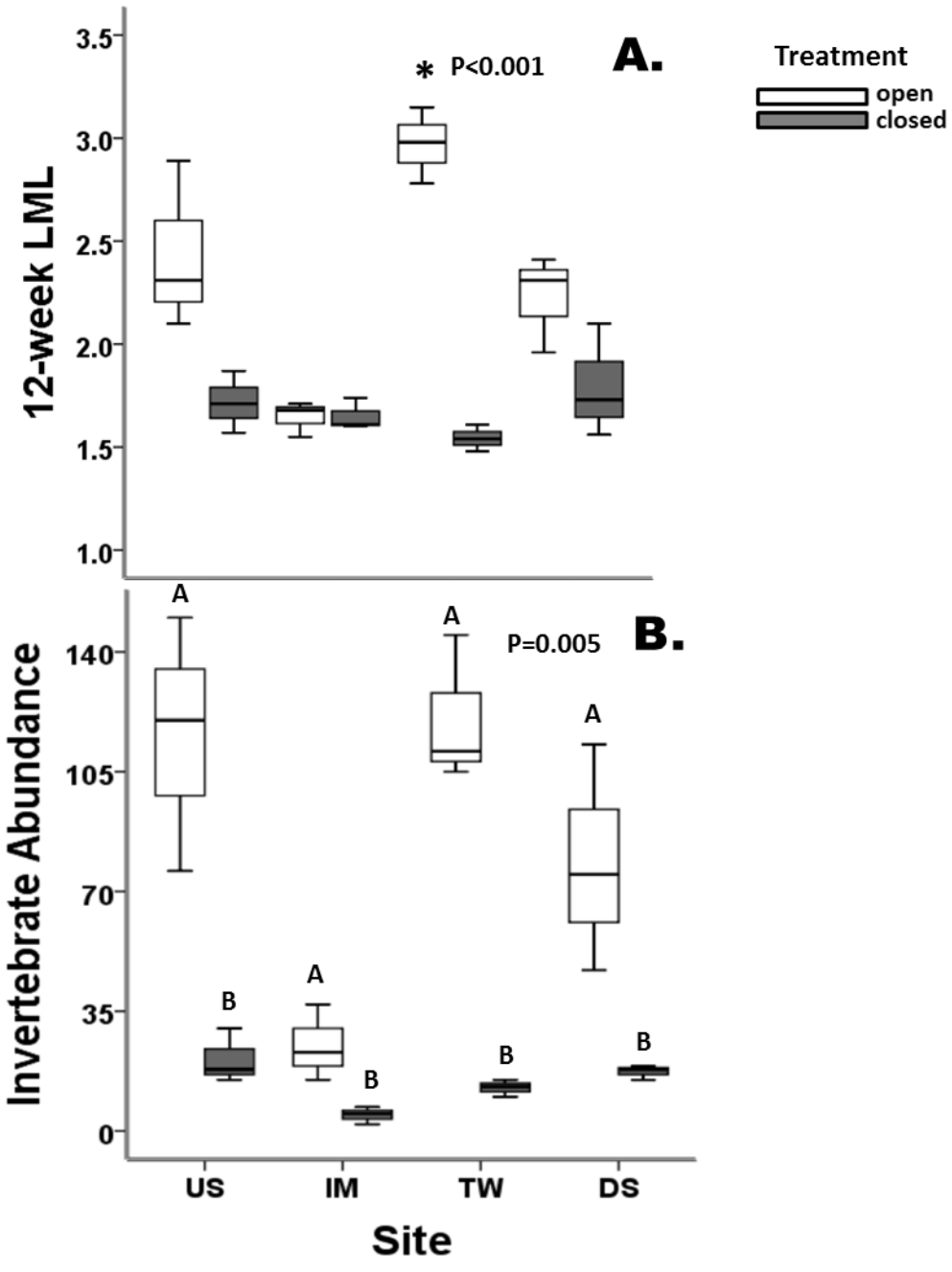

A.
* P<0.001
Treatment
open
closed
12-week LML
3.5
3.0
2.5
2.0
1.5
1.0
B.
P=0.005
Invertebrate Abundance
140
105
70
35
0
A
B
A
B
A
B
A
B
US
IM
TW
DS
Site

Figure 8:

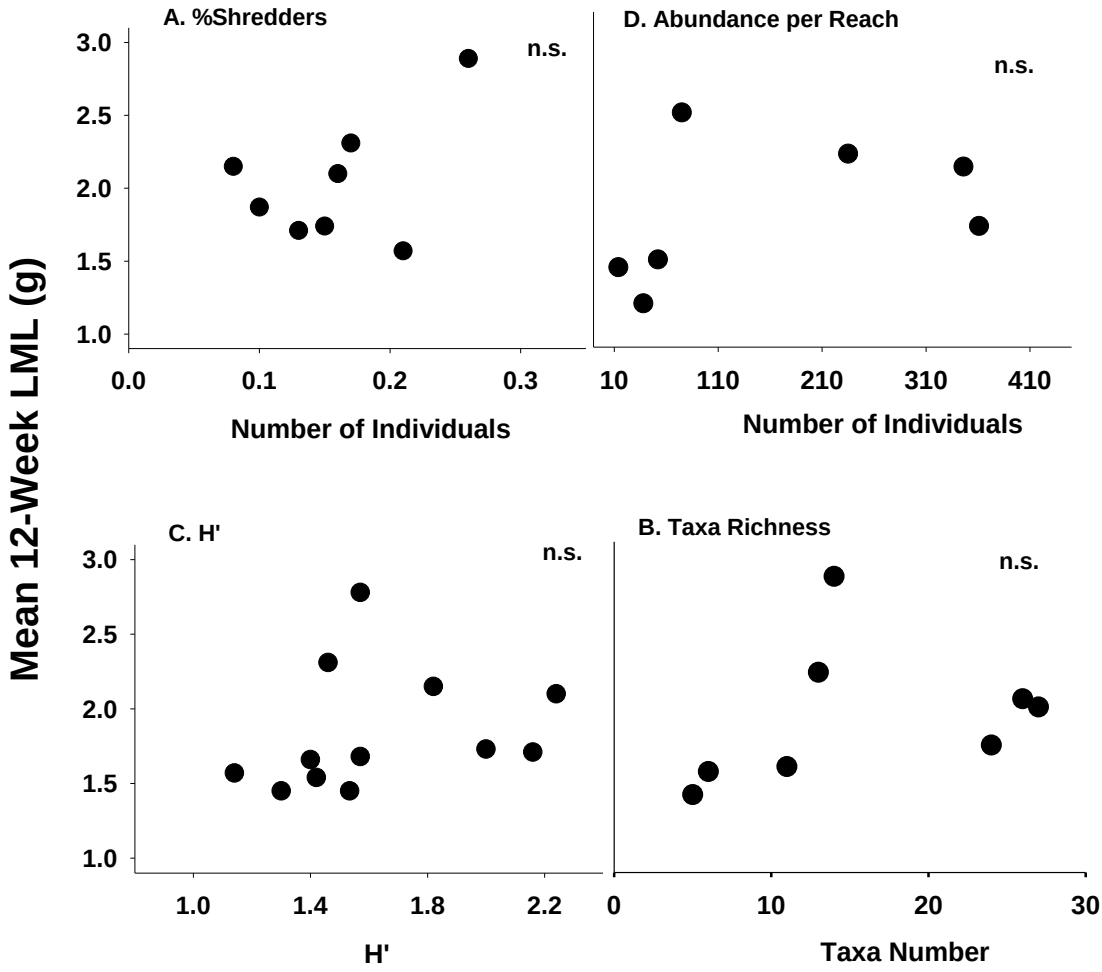

A. %Shredders
n.s.
3.0
2.5
2.0
1.5
1.0
0.0
0.1
0.2
0.3
Number of Individuals
D. Abundance per Reach
n.s.
10
110
210
310
410
Number of Individuals
Mean 12-Week LML (g)
C. H'
n.s.
3.0
2.5
2.0
1.5
1.0
1.0
1.4
1.8
2.2
H'
B. Taxa Richness
n.s.
0
10
20
30
Taxa Number

Figure 9:

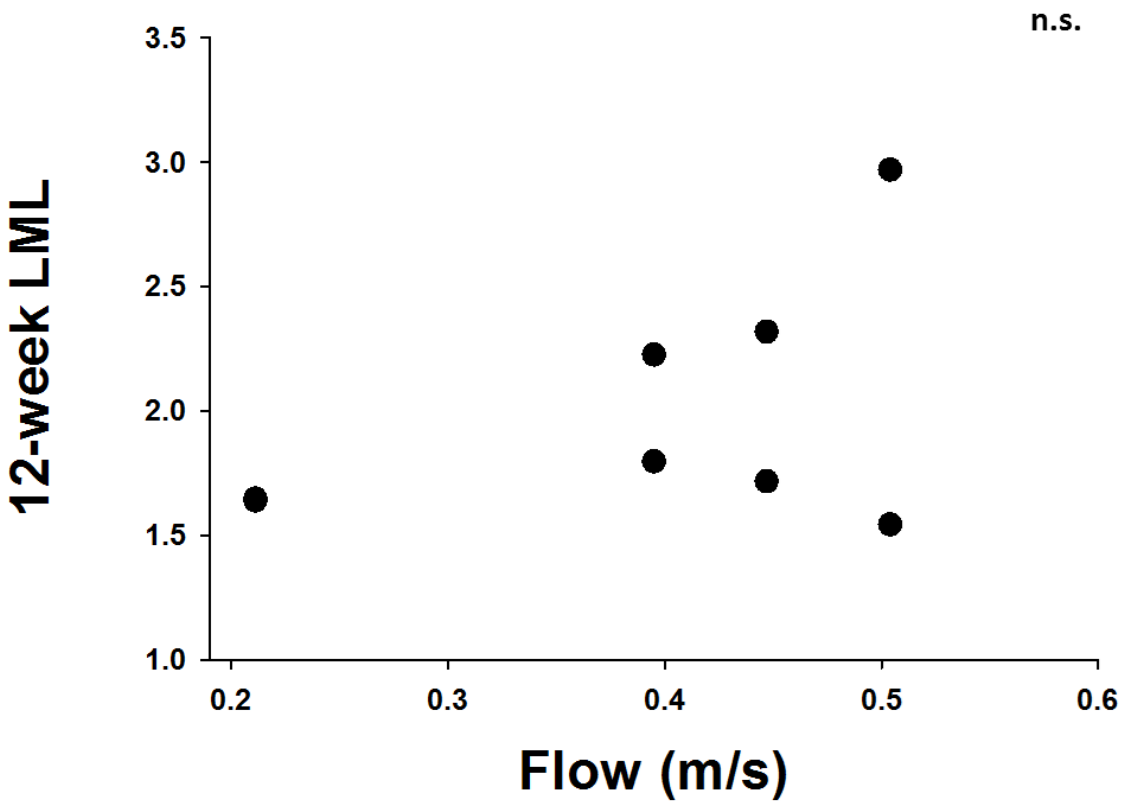
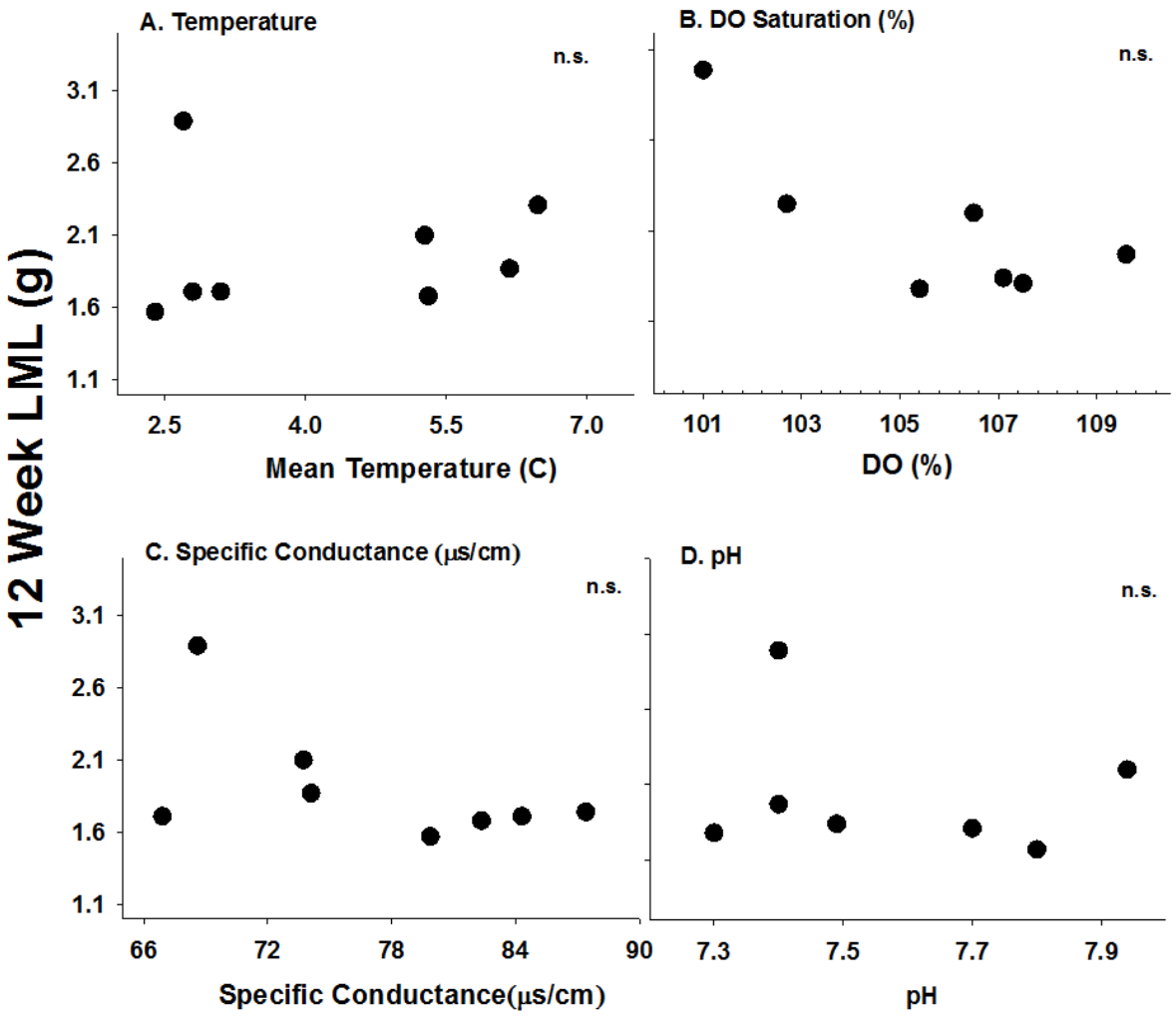


Figure 10:


A. Temperature
n.s.
B. DO Saturation (%)
n.s.
12 Week LML (g)
3.1
2.6
2.1
1.6
1.1
2.5
4.0
5.5
7.0
Mean Temperature (C)
101
103
105
107
109
DO (%)
C. Specific Conductance (µs/cm)
n.s.
D. pH
n.s.
66
72
78
84
90
Specific Conductance(µs/cm)
7.3
7.5
7.7
7.9
pH

Vita

Beverly Elyse Russing was born in Boone, NC on May 24th 1988 to Caroline and Michael Ivey. She attended high school in Fayetteville, NC and graduated from Fayetteville Academy in 2007. In the spring of 2014 she was awarded a Bachelor of Science degree from Appalachian State University with a focus in environmental science and conservation biology. In the fall of 2015, she accepted an assistantship at Appalachian State University to pursue a Master of Science degree. Upon completion of her Master's degree, Beverly hopes/intends to find employment with a program in community outreach.